Integrated water management, ecosystem services and food security - scoping study

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1 Introduction

History

Since the 1960s an important paradigm shift in science can be observed in our understanding of ecology, how it works, its complexity and dynamic internal and external relationships. These external relationships refer to what our natural environment and the ecosystems surrounding us mean for society. Parallel to the development of ecological system dynamics increasing attention was also being paid to the co-evolution of natural and human systems in environmental and ecological economics. An example is the work of Holling who introduced the concept of functional response in population ecology and other ideas such as ecosystem resilience and adaptive capacity.

The way this was conceptualized and operationalized was through the functional performance of ecosystems. Humanity needs ecological systems, ecosystems in short, to survive. The organic and inorganic components and processes within ecosystems provide functions, which benefit human beings through the social and economically valuable 'goods and services' these biophysical functions provide. Examples include clean drinking water and food needed for humanity to survive, but also less obvious goods and services perhaps such as human health protection as a result of clean air, climate regulation and UV filtering by the ozone layer.

The concept of ecosystem services became even more popular after the Millennium Ecosystem Assessment (2005), which introduced the term to underline the inextricable linkages between biodiversity, ecosystems and human well-being. Ecosystem services are 'the benefits people obtain from ecosystems' and by putting ecosystem services central in the debate on nature conservation, the Millennium Ecosystem Assessment stresses the societal benefits of nature conservation and the need to align conservation and development goals. Important ecosystem services are food, fuel, timber and water provision, carbon storage and sequestration, soil formation, climate and disease regulation, aesthetic benefits and spiritual values (MA 2005).

Since nature is the underlying asset from which a substantial share of the total ecosystem services are produced (MA 2005), ensuring the provisioning of ecosystem services requires that biodiversity is well protected and that ecosystems are well-managed. With regard to the current status of biodiversity protection the UN (2010) states that: "As a consequence of human actions, species are being lost at a rate estimated to be 100 times the natural rate of extinction. In the past century, 35% of the mangroves, 40% of the forests and 50% of the wetlands have been lost (..) the 2010 biodiversity target has not been met at the global level. None of the 21 sub-targets accompanying the overall target have been achieved (..). Action is urgently needed to avoid reaching tipping points or critical thresholds that will lead to irreversible loss of biodiversity and ecosystem services, with dangerous consequences for human well-being" (UN 2010).

Demand and supply

The basic idea behind ecosystem functions providing ecosystem services is simple and appealing from an economic point of view as it reduces the issue to a question of supply and demand. The environment supplies through its functionality goods and services which are in demand by human beings. We need clean water, air and food, all provided 'free of charge' by our planet, but also value nature as a place to enjoy recreational activities, find inspiration, or feel good, knowing that there is a place for wildlife with which we co-exist on this planet.

If demand for these goods and services is higher than supply and becomes competitive for the limited available goods and services, there is scarcity. This competing demand strips down the issue of sustainable environmental management, meeting demand now and in the future and carefully balancing social, economic and ecologic interests at the same time, to a fundamental economic question. This idea goes back to theories of classical economists like Thomas Malthus and David Ricardo in the 18th century who expressed their concerns about the limited availability of one of the first natural resources considered in economic thinking, namely land for agricultural production. Malthus was primarily concerned about absolute scarcity of land, so the available amount of land, while Ricardo thought it would be the quality of the natural resource land that imposes limits to economic growth.

Even though there exists no market for many ecosystem services where they are traded as a result of supply and demand, they do have value and are not really 'free of charge' or 'cost-free' so to say, as there are almost always opportunity costs involved. For instance, their current use is at the expense of their future use as is the case for non-renewable energy resources such as oil and gas, or their use by one particular group is at the expense of another group of people as is the case when we have overcrowded common pool resources like natural parks or beaches, or discharge wastewater in a river impairing recreational fishing opportunities.

The trade-off between economic development on the one hand and environmental conservation on the other hand based on an assessment of the supply and demand of ecosystem goods and services is nowadays generally accepted by both natural and social scientists. In the natural sciences important research questions relate to improving our understanding of the bio-geochemical structure and processes underlying ecosystem functions. For example, what is the role of biodiversity in ecosystem processes, how do nitrogen cycles induce environmental change and affect the provision and quality level of ecosystem services.

Ecosystem services and food security

Gordon et al (2010) summarize the tradeoffs that have arisen because of the exclusive focus of water management on agricultural food production, identifying three strategies through which agricultural water management can use a more integrated approach:

a. Improving water management practices on agricultural lands;

b. Better linkage with management of upstream and downstream aquatic ecosystems; and

c. Paying more attention to how water can be managed to create multifunctional agro-ecosystems.

Rosegrant et al. (2002) underline the importance of environmental water uses in their report on future world water demand and global food production by stating that: "(..) environmental uses of water, which may be key to ensuring the sustainability of the Earth's water supply in the long run, often get short shrift". By 2025, "water withdrawal for most uses (domestic, industrial, livestock) is projected to increase by at least 50%. This will severely limit irrigation water withdrawal, which will increase by only 4 percent, constraining food production in return (Rosegrant et al. 2002)". They go on to argue that sustainable water management requires more efficient use of water in agriculture, increased access of domestic users to piped water and increased environmental flow (Rosegrant et al. 2002). Also, in order to safeguard food production, attention should shift to green water management (soil moisture) and the efficiency of water use.

This concept note will (1) discuss why shifting attention from agricultural water management to integrated water management might help ensure food security and (2) demonstrate what this shift implies for the management of water resources at global scale.

With regard to the first question, Table 1.1 summarizes the different contributions that agricultural and non-agricultural ecosystems make to food production. The distinction between agricultural and non-agricultural ecosystems is useful since it underlines the fact that crucial ecosystem services might come under pressure when water management ignores the importance of environmental flow. For example, pest regulation and erosion control might be mostly related to sustainable management of agricultural ecosystems, but natural hazard and climate regulation, pollination, nutrient cycling and genetic diversity depend on the management of the broader ecosystem, including the non-agricultural parts, as well.

Table 1.1 Contribution of ecosystem services to food production (FAO 2008)

Provisioning	Regulating	Supporting	Cultural
• Food and nutrients • Fuel • Animal feed • Medicines • Fibres and cloth • Materials for industry • Genetic material for improved varieties and yields • Pest resistance	• Pest regulation • Erosion control • Climate regulation • Natural hazard regulation (droughts, floods and fire) • Pollination	• Soil formation • Soil protection • Nutrient cycling • Water cycling	• Sacred groves as food and water sources • Agricultural lifestyle varieties • Genetic material reservoirs • Pollinator sanctuaries

Rockstrom et al. (2009) has indicated that several of the ecosystem services are under strong pressure, and that failure to address these imbalances might result, or already have resulted, in total system collapse. The uncertainties surrounding the ecological, hydrological and atmospheric processes underlying ecosystem service provisioning are large however and it is difficult to say whether the current system is relatively stable or close to system collapse. Many scholars are arguing that to deal with these uncertainties it is important that agricultural policy, land use planning and water management become more adaptive and use a more precautionary approach (see for example Ericksen 2008).

In this note we will elaborate how integrated water management and land use planning can contribute to a more resilient global food production system that is adaptive to environmental and climate change. We will do this by elaborating the second question, the implications of integrated water management for the management of water and land. Water and land are mentioned jointly, since integrated water management implies that water resource management is integrated with the management of soils, forests, wetlands, mangroves and other land uses that are directly linked (GWP 2000). Integrated water management also implies integration across scales, involving local, regional and international users in basin level decision-making, explicitly recognizing the tradeoffs that may arise between the different levels and between up and downstream water use.

The ecosystem services approach

In discussing the added value of integrated water management, and in a broader sense attention for ecosystem services governance, we will use Figure 1.1 developed by Daily et al (2009) to describe the difficulties associated with using an ecosystem based approach. Roughly, Daily et al. (2009) distinguish three domains where changes are needed, i.e. the ecosystem domain (how do decisions affect ecosystems and the provisioning of ecosystem services at different scales), the economic domain (how do ecosystem services contribute to human well-being and how is this reflected into existing incentive structures) and the governance domain (how are ecosystem values embedded in institutions and other governance mechanisms and how is this translated into decision-making at multiple scales).

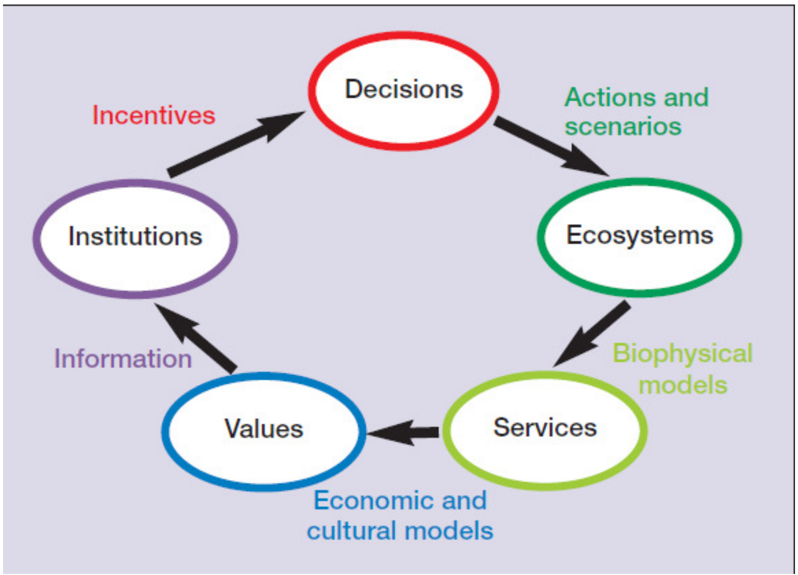

Figure 1.1 Integration of ecosystem services into decision-making (Daily et al. 2009)

We will illustrate each of these domains with examples from the literature and from our own research to highlight the types of issues that arise when implementing integrated water management. From our experience, the issues that arise are rather similar around the world, although the urgency of climate change, food production and water scarcity clearly differs a lot.

Integrated water management effectively requires that the problems arising in each of the domains shown in Figure 1.1 are addressed. For example, without a better understanding of the spatial linkages between alternative land and water uses, water managers cannot improve the efficiency of water allocation at basin scale. Also, they need to have insight into the potential impacts of climate change, the environmental and non-environmental impacts of increased environmental flow and understand how water re-allocation might contribute to human well-being and what tradeoffs may arise at basin scale. If the political will is lacking, however, to incorporate these values into decision-making and change the existing institutions and incentive structures in order to better reflect the scarcity value of water resources in decision-making processes at multiple scales, more information and insight into the ecosystem and the economic value of its services will be of little use.

In the following Sections, we will further discuss these domain-specific issues (i.e. ecological, economic and governance domains) and come up with practical suggestions about the possible actions to facilitate integrated water management.

2 The ecosystem domain

The ecosystem domain of Daily's framework (see Figure 1.1) refers to the ecological and biological processes underlying the provision of ecosystem services. In this domain, the ecological trade-offs and synergies in ecosystem services benefitting food production can be identified.

Ecological scale

One of the dimensions that play an important role in identification of potential synergies is the **scale** at which potential interventions can occur. For example, Integrated Water Resource Management (IWRM) and Land Use Planning (LUP) perceives ecosystems at the larger landscape scale of catchments at ~10-1000 km . This may reveal improvement options for "large-scale" ecosystem services such as watershed services and carbon sequestration. At this larger, catchment scale the most promising synergies are to be found where the surrounding landscape ensure continued food productivity. This lies in the provision of sufficient water and the safe/guarding of soil quality and nutrient availability. Focus should therefore be on supporting services, hence on the interactions between intermediate services and their optimisation. Small-scale ecosystem services (pollination, natural pest control by e.g. green-blue veining) may well operate at the smaller field or field-margin scale (10-100 m) and hence be ignored in IWRM assessments. The synergy between landscape planning for biodiversity conservation and food production enhancement may well rest in spatial allocation of habitat mosaics and green-veining.

Synergy

Another important dimension in the ecological domain of the ecosystem services approach is the extent to which provision of specific ecosystem services is **complements or substitutes** the provision of other ecosystem services. While in traditional analysis landscapes are often managed to optimize the provision of single ecosystem services, such management may lead to negative impacts on other services. For example, land planning and management has focused on maximizing rational food production after the Second World War, whilst ignoring negative externalities that in the long run may affect food production adversely (e.g. loss of precious top soil and salinisation through poor irrigation practice). The innovative trade-off analysis between ecosystem services is recently done based on the analysis of spatial congruence of mapped ecosystem services (Swallow et al., 2009; Raudsepp-Hearne et al., 2010; Willemen et al., 2010a). Trade-offs that result from changes in ecological or socio-economic processes may not be uncovered this way. Bennett et al. (2009) indicate that it is essential to understand the processes underlying interactions between ecosystem services for analyzing possible conflicts and synergies in ecosystem service provision for different management alternatives. A number of studies have identified clusters of ecosystem services that often appear together with known interactions. Such 'bundles of ecosystem services' are often found at specific places in the landscape and result from interactions between ecosystems processes and management (Raudsepp-Hearne et al. 2010). Typical examples are the overlap between areas with a strong role in the provision of watershed services while simultaneously supporting biodiversity and carbon sequestration (Verweij et al. 2008).

An approach that optimizes management taking into account interactions between ecosystem services is likely to have many benefits. In a recent paper by Egoh et al. (2010) it was shown that conservation plans in the Little Karoo region of South Africa can be made far more efficient by selecting areas for both biodiversity and ecosystem services at no or minimal additional costs. Another example of this is provided by the US-based Natural Capital Project (www.naturalcapitalproject.org) which produced strong evidence that accounting many comprehensive conservation strategies can generate both a wide range of ecosystem services, economic cost and benefits, in addition to stringent biodiversity targets. Both scenario studies (Nelson et al., 2009) and optimization studies (Polasky et al., 2008) are used to make the concept of ecosystem services operational within a policy context (see coming Sections).

3 The economic domain

The economic domain of integrated water management in relation to ecosystem services concentrates around the question how ecosystem services contribute to human well-being and how is this reflected into existing incentive structures. These two subjects will explained accordingly in the subsequent Sections.

Economic valuation of ecosystem services

The literature regarding the implementation of analyses of economic valuation of biodiversity and ecosystem services is expanding rapidly. This literature provides much of the theoretical framework underpinning the ecosystem services concept. In particular it highlights the way in which stocks of natural assets yield flows of services which either directly or in combination with further inputs (e.g. manmade and human capital, etc.) produce goods and services which generate benefits realised by people both through consumption and non-consumptive use and through non-use existence and bequest values. However, while the literature provides a firm theoretical underpinning for the ecosystem services approach, to date there has been little practical implementation of these principles within applied decision making. One of the major issues which have to be addressed to facilitate this implementation is that the ecosystem services approach requires the seamless integration of natural and social science knowledge of the operation of natural resources within the process of generating human wellbeing. The state of play is that the natural sciences often do not yield the forms of information flows required for economic analyses.

The focus of virtually all valuation exercises is the **flow** of ecosystem goods and services delivered by natural assets. However, this overlooks the vital importance of maintaining the **stocks** of those assets at sustainable levels, also referred to as 'insurance value' of maintaining ecosystem resilience (Mäler et al., 2009; Mäler, 2008; Walker et al., 2010). Many environmental assets exhibit local or global irreversibility which means that the running down of such stocks penalise future generations, diminishing their opportunity set in ways which contravene the principles of sustainability (Gerlagh and van der Zwaan, 2002; Hoel and Sterner, 2007; Sterner and Persson, 2008; Pascal et al., 2009). Furthermore, several of these assets exhibit threshold levels below which the rate of stock decline markedly accelerates in ways which threaten resource crashes (Lenton et al., 2008; Rockström, et al., 2009) (e.g. the running down of species populations). From an economic perspective such stocks exhibit increasing marginal restoration costs beyond these thresholds (put simply the cost of restoring each unit of such a resource begins to rise once a threshold has been breached). This problem is exacerbated in cases of hysteresis where the breaching of thresholds causes ecosystems to flip into hyper-degraded states which necessitate entire cessation of any economic driver and relatively extreme costs in order to even prepare for any restoration of the asset (e.g. the euthropication of certain shallow lakes). Finally in extreme cases, breaching of thresholds causes tipping point effects from which no restoration is possible. The necessary theoretical framework for analysing such stock problems has only recently been formulated through the notion of "Comprehensive Wealth" (Dasgupta and Mäler, 2000; Arrow et al., 2007; Mäler et al., 2008; Dasgupta, 2009).

Incentive structure

Water management in Europe is globally heralded as a textbook example of integrated water policy. Integrated water policy is defined here as the 'coordinated management of water, land and related resources in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems (GWP 2000)'. The European Water Framework Directive (WFD, 2000/60/EC), adopted in 2000, pays explicit attention to hydrological, chemical, ecological, economic and social objectives and the coordination of water, land and ecosystem management across issues and scales. Water managers should not only 'reduce, or phase out, chemical pollution', but water bodies should also reach 'a good ecological status' whilst ensuring 'a cost-effective selection of measures' and avoidance of 'disproportionate costs' (EC 2000a). In addition, 'floods and droughts should be mitigated' and river basin management plans should seek to 'reduce flood risks' while 'fairly sharing' responsibilities at transboundary scale (EC 2007a, b). To ensure coordination among water bodies, European water management is organized around river basins and authorities collaborate in transboundary river basin management plans.

Economic instruments, also called market-based, incentive-based or pricing instruments, form an integral part of European water policy. The WFD states, in Art. 9, that member states should 'take account of the principle of recovery of the costs of water services, including environmental and resource costs' and that water pricing policies should provide 'adequate incentives' for sustainable water use by 2010 (EC 2000a). In addition, the European Commission published several communications promoting the use of water pricing and other market-based instruments to reduce over-extraction and diffuse (nitrate) pollution of surface and groundwater (EC 2000b), to promote water saving and avoid inefficient surface and ground water use (EC 2007b) and to meet the objectives of the WFD in general (EEA 2005a, EC 2007e). The evaluation of the first stage implementation of the WFD suggests, however, that attention to economic instruments is lacking and that member states will have a hard time meeting the requirements of Art. 9 (EC 2007c,d). In fact, the European Commission has declared promotion of the use of economic instruments in river basin management plans a priority, considering that full exploitation of economic instruments will contribute to 'truly sustainable water management' at basin scale (EC 2007c).

The reason why economic instruments are hardly being taken up in European river basin management plans remains an open question (Kampa et al. 2009). Experiences from around the world show that economic instruments can effectively reduce water pollution (Rolfe and Windle 2009; Selman et al. 2009) and that they can play an important role in enhancing sustainable water use (Griffin 2006; Winpenny 2005). In fact, the limited European experiences with water pollution charges are positive (OECD 1997), and suggest that economic instruments can be a cost-effective water management tool (EEA 2005a,b). Given that institutional and environmental factors determine instrument effectiveness to a large extent (Andersen 2001), accounting for these factors when analysing the potential role of economic instruments in European water policy is important. Also, transaction costs and behavioural factors should be included in the analysis, as they influence instrument effectiveness and efficiency too (Cummings et al. 2004). Furthermore, there may be social and political reasons for water managers to prefer traditional command-and-control instruments (Kirchgässner and Schneider 2003), which need to be addressed. Thus, whether economic instruments are an efficient and effective option is not always self-evident, and a comprehensive evaluation of instrument effectiveness is required to determine the effects. Also, the effect of introducing economic instruments on poverty is also an under-researched area.

Practical guidance

From the economic domain, several practical policy recommendations result. First, **experiment with payments for ecosystem services** in integrated supply chain management. Despite the absence of markets and prices for many ecosystem services, it is in some cases an observable fact that they have value and that they are assigned financial and economic values in policy and decision-making processes. Farmers get, for example, compensated for agricultural wildlife management. The compensation price levels are based on actual labor costs and production losses and, depending on the exact activity, a mark-up between zero and twenty percent to encourage farmer participation. These compensations were introduced in the Netherlands after adoption of the so-called Relatienota in 1975, to account for the negative externalities of modern-day agriculture and encourage the production of positive externalities such as biodiversity and landscape amenities. Under the umbrella of the European Common Agricultural Policy, other European Member States such as the UK, Belgium, France, Italy, Ireland and Finland have introduced a wide variety of agri-environmental schemes to comply with the European Rural Development Regulation since 1999.

Second, **broaden the definition of water services** (including regulating services of the ecosystem). Interpreting water services in a broader sense than currently is the case is expected to be beneficial in the long term to achieve more sustainable levels of water use. Expressing nonmarket ecosystem goods and services in monetary terms – as long as we do this in a sound and robust way – provides an important signal to policymakers that our natural environment has a value and that using it in an unsustainable way comes at a cost. Money speaks louder than words, especially when dealing with companies whose business it is to make money. Introducing financial incentives to change the way they operate and use the environment as if the ecosystem services it provides are free of charge can be an important complementary policy to traditional command and control to change people's behaviour. For example, making companies pay for permits issued by the government to pollute and allow some degree of market functioning where these permits can be traded to stimulate more efficient water use. Making money talk means that money does the talking within the environmental boundaries imposed based on safe minimum standards or precautionary principles.

Third, focus on **agricultural price incentive** for low water intensity crops. Water pricing is often dismissed in policy and decision-making on incorrect grounds, namely that water use is non-responsive to water pricing. Water allocation and water pollution rights are very much dominated by traditional technical standards and engineering solutions. We have a much better understanding already of the values underlying water services. One has to be careful to use some of these ball-park estimates to fix price levels, but if you look closer, some remarkably consistent results are found with important policy implications. For example, the value of water for many low value crops is universally low. When used for high-value crops, the value of water can be much higher, sometimes in the order of magnitude to the value of water in domestic and industrial uses. The value of water for domestic use is, not surprisingly, always highest, whereas the value for environmental purposes such as environmental flows, maintenance of wetlands, wildlife habitat vary widely, but typically fall in between the agricultural and domestic values. The most important conclusion here is that water users do react to water pricing by reducing their demand if the price increases. It is often a misunderstanding that an inelastic demand for water means that water use is completely unresponsive to changes in water prices.

4 The governance domain

"For the last 25 years, prescriptions of the water policy literature have centred around two themes. The first is that 'the watershed' is the appropriate scale for organizing water resource management [...]. The second is that since watersheds are regions to which political jurisdictions almost never correspond, and watershed-scale decision making structures do not usually exist, they should be created." (Schlager and Blomquist 2000:1)

Issues surrounding river-basin organizations

With regard to the creation of river basin management authorities, governance scholars raise several issues in relation to the institution and operation of river-basin organizations (Imperial and Hennessey 1999:5). First, they suggest that the **boundaries** of river basins are not necessarily so clear or "natural" (Schlager and Blomquist 2000:12–17). The idea of "the" river basin suggests a certain simplicity, which in reality does not exist as river basins are connected (sometimes by human intervention) and nested. This means that defining the boundaries of a basin requires choice, and this implies a role for politics. To quote Schlager and Blomquist (2000:15–16): "Boundaries are multiple, overlapping, and often contested because people experience and attempt to deal with a host of problems and opportunities that vary in scale from the local to the regional. Drawing boundaries is the first step in determining who decides and how and with what effects. Different boundaries imply different decision makers and different effects." Some communities may lose local control, whereas others more may gain more control. Especially those who benefit from the current boundaries may object to reshaping the boundaries.

Second, governance scholars draw attention to the fact that after founding a river-basin organization, it becomes necessary to formulate **decision-making arrangements**. Two available alternatives mentioned in this respect are consensus and elite decision making. Consensus decision making draws the risk of gridlock, whereas elite decision making may result, among other things, in the exploitation or oppression of minorities (Schlager and Blomquist 2000:17–18) or in non-implementation of decisions if influential stakeholders have not been involved (cf. Ridder et al. 2005). In practice, decision-making arrangements are a mixture of these options. Imperial and Hennessey (1999:27–35) suggest that, in designing decision-making arrangements, the emphasis should be on regular meetings between the partners, reduction of power and information asymmetries, minimizing the risk of strategic behaviour from participants, and enabling (bureaucratic, legal, professional, and political) accountability.

Third, there are issues of **authority**, that is, issues of tasks and responsibilities for the new organization (Schlager and Blomquist 2000:20–23). Governance scholars warn that large unitary river-basin authorities are just as susceptible to "bureaucratic pathologies" as any other bureaucracy (cf. Biswas 2004). Schlager and Blomquist (2000) make the point that institution building tends not to follow a pre-established design but can be better described as a patchwork. In composing the patchwork, environmental concerns are far from dominant. Instead, economies of scale, the division of skills across organizations, the costs of coordination, and issues of culture and political identity are said to be more important (Schlager and Blomquist 2000:20–23). Interestingly enough, governance scholars suggest that a patchwork of institutions at various overlapping levels may not only be more feasible, but also more desirable from an environmental perspective than a unitary river-basin authority because of the possibility for reorganizing the patchwork according to the necessary task.

The idea of addressing water issues at the basin scale has been influential in practice. In a worldwide survey, Dinar et al. (2005) found hundreds of transboundary basin organizations. Supporting the idea of institution building as a patchwork, governance scholars find that the pattern of institution building reflects the importance of governance considerations (politics, institutions) vis-à-vis environmental goals. For instance, Schlager and Blomquist (2000:4; quoting others) suggest that most American examples of river-basin organizations reflect their current institutional contexts, in the sense that they usually do not have formal decision-making powers and sanctioning authority. Conca et al. (2006) analyzed a worldwide set of 62 transboundary river agreements. They found (Conca et al. 2006:271–282), among other things, that many agreements do not include all states in a basin and that transboundary agreements are concentrated in basins with a tradition of cooperation. They also found that hegemonic states are more likely to participate in such agreements, and that agreements tend to express both the need for responsible management and state rights. Finally, their data tentatively suggest that the content of such agreements depends on power relations between the signatories, with the agreements stressing principles that are advantageous to hegemonic states.

Imperial and Hennessey (1999:22) suggest that the "collaborative capacity" of organizations operating in a basin depends on their capacity for problem solving, slack resources, and stable sources of funding. Furthermore, concurrent with Ostrom and Janssen (2004), their analysis of six U.S. cases suggests that a high collaborative capacity may be correlated to the presence of an "institutionally rich environment" (Imperial and Hennessey 1999:22), meaning that multiple organizations have overlapping roles to play in water management. This further supports the case for polycentric governance systems.

Box 1. What is governance of ecosystem service?

In the 1990s, scholars seized on the term 'governance' to make better sense of the situation that had arisen in many countries after the 1980s, when 'big' government had retreated under the pressure of neo-liberal reformers like Ronald Reagan and Margaret Thatcher (Rhodes 1996). Some have suggested that the turn from government to 'governance' was driven by big business and its desire to weaken the regulatory powers of the nation state (e.g. Swyngedouw 2005). Others offered more prosaic explanations, including the financial crisis of the state in the 1970s and 1980s, and the associated ideological shift towards the market and 'new public management' (Pierre and Peters 2000). What attracts social scientists to the term 'governance' is its ability to 'cover the whole range of institutions and relationships involved in the process of governing' (Pierre and Peters 2000: 1).

Clearly, 'governance' is not the same as government: while government centers on the institutions and actions of the state, the term governance allows non-state actors such as businesses and civil society to be brought into an analysis of societal steering. Governance is also not the same as governing. 'Governing' refers to those social activities which make a 'purposeful effort to guide, steer, control, or manage (sectors or facets of) societies' (Kooiman 1993: 2; Rosenau 1992: 4). 'Governance', on the other hand, describes 'the patterns that emerge from the governing activities of social, political and administrative actors' (Kooiman 1993: 2). It concerns 'the ways and means in which the divergent preferences of citizens are translated into effective policy *choices*, about how the plurality of societal interests are transformed into unitary action and the compliance of social actors is achieved' (Kohler-Koch 1999: 14).

In the governance literature there is still relatively much disagreement about what governance *is*. There is however relatively widespread agreement on a number of basic points. First and foremost, most scholars seem to associate governance with a decline in central governments' ability to steer society (i.e. the narrower interpretation above). In this respect, larger shifts in governance can be observed in many European countries. Figure 2 below signifies these shifts. Second and more controversially, governance and government are often (and most notably in the older political science literature) regarded not as discrete entities, but two poles on a continuum of different governing types (Finer 1970). Third, that there is no governance 'theory' or even proto theory (Pierre and Peters 2000). Many scholars use the term governance to problematize the relationship between the state, the market and civil society i.e. how and through what mechanisms is the state attempting to adapt to changes in its external operating environment? Thus, under a 'government' approach, it is commonly assumed that society is steered from the centre (normally by the state), whereas in a 'governance' model, 'society actually does more self-steering rather than depending upon guidance from government' (Peters 2000: 36).

	Markets ***PPS and Privatization***	
Higher and lower levels of governments ***Europeanization, regionalization***	**《 From Nation State to ... 》**	**Communities** ***Self governance, participation***
	Courts ***Judicialisation***	

Source: Huitema 2005

Effectiveness of the river-basin approach

There is little empirical evidence for the effectiveness of the river-basin approach, either in its monocentric form (unitary river-basin authorities) or its polycentric form (collaboration at the basin scale), in the literature discussed here. Dinar et al. (2005:4–5, 15) suggest that basin-level governance institutions are a necessary but insufficient condition for successful resource management, meaning that the absence of such institutions will lead to the failure of management but their presence does not necessarily lead to success. They suggest the responsiveness to sub-basin stakeholders is one of the more important factors in explaining institutional effectiveness. Imperial and Hennessey (1999:23) provide evidence of environmental improvements as a consequence of collaborations at the basin level, which resulted from a shared set of regulations. Additional benefits that they find are the emergence of economies of scale in dividing tasks across government bodies, greater citizen involvement, and learning in the form of increased levels of trust between organizations and greater success in lobbying higher-level authorities. Some authors contest the added value of the approach, however, and suggest that the focus on the geographic boundaries in explaining governance performance has obscured other important variables, such as population growth, international relations, and regional economic cooperation (e.g., Pimentel at al. 1994, Allan 2003, Wirkus and Böge 2005, Mostert 2009).

In all this, we have to remember that we cannot design institutions from scratch because of opposition by those who have vested interests in the present institutions. Moreover, it might be too intellectually challenging as it is very hard to predict with any degree of certainty how completely new institutions will work out in practice. Institutional design calls for careful experimentation and learning from experience.

Adaptability of ecosystems and social systems

With regard to issues relating to adaptability, the unpredictability of ecosystems and their response to human interferences have been major tenets in the literature on resource management in the past decades. Dryzek (1987:28–33) suggested that this unpredictability is due to, among other things, the complexity, non-reducibility, spontaneity, variability, and collective quality of ecosystems. Social systems exhibit similar qualities, and increasingly so as the web of connections between countries, their economies, and governments grows denser and denser because of globalization (Young et al. 2006). This makes the management of "social–ecological systems" (Berkes and Folke 1998) a daunting challenge.

Following Young et al. (2006), we pose that adaptation refers to the process of structural change in response to structural circumstances. Effective adaptation results in adaptedness, meaning that a certain dynamic structure is effective in dealing with its current external environment. Adaptability is about the capacity to adapt to future changes in the environment of a particular system. Adaptive governance refers to the totality of interactions, by private and public actors, to achieve adaptation and to enhance adaptability.

On the one hand, adaptation differs from adaptability in that it implies change with an eye to current ecological conditions and it therefore implies the implementation of a set of measures to deal with them. On the other hand, adaptability is at a somewhat higher abstraction level and is about 'meta characteristics' of a social system. Folke et al. (2005: 444) suggest that in a social-ecological system with high adaptability, the actors have the capacity to reorganize the system within desired states in response to changing conditions and disturbance events. They (ibid.: 444-448), along with others such as Lee (1999), indicate that adaptability is about various 'meta-characteristics', including flexibility (seeing policies as experiments, making use of crises), a new role for scientists (policy science versus research science), and about organizational learning (reflection upon goals, interests, instruments, etc.). In addition, they pose that adaptability requires a range of other factors, such as a holistic approach to ecosystems, good leadership, voluntary self-organizing and self-enforcing institutions, public participation and deliberation, a diverse and redundant set of institutions, high trust social networks and places with social memory.

Box 2: Centralization vs decentralization in integrated water management

Integrated water management requires basin-level coordination, but all over the world natural resource management is being decentralized to local communities to enhance local participation and increase effectiveness (Mansuri and Rao 2004). In India, community-based soil and water conservation (SWC), or watershed development, has long been one of the main programs for rural development, annual expenditure reaching up to 500 million USD (Kerr et al. 2002). Investments aim to improve the productivity of dryland agriculture by increasing soil moisture and recharging groundwater aquifers for supplementary irrigation (Bouma and Scott 2007). Typically, SWC investments consist of small dams, earthen bunds, trenches and village ponds that aim to reduce the speed of surface water run- off and allow for more of the rainfall to be locally absorbed. To optimize returns, investments are planned at the scale of the micro-watershed (500-1000 ha) and implemented at village level. Although several studies have shown that investments are welfare enhancing at village level (see for example Kerr et al. 2002), up till recently no assessments of the potential impacts on downstream users were made.

A recent publication by Bouma et al. (2011) indicates, however, that upstream investments in soil and water conservation may have negative impacts at basin scale. Using a combination of hydrological and socio-economic data, the paper evaluates the basin wide welfare impacts of rainwater harvesting in a water scarce, semi-arid basin by assessing the increase in (groundwater irrigated) agricultural production upstream and the decrease in (surface water irrigated) production downstream, and accounting for rainwater harvesting investment costs. The analysis shows that only when upstream producers generate double the value of what downstream producers would have produced otherwise are upstream benefits sufficient to compensate downstream losses and pay-back the investment costs. The findings imply that better consideration is needed of the downstream externalities in decisions on rainwater harvesting projects. Although the findings are robust to changes in land use and prices, further research is required into the water balance of irrigation systems and the potential wider welfare effects of soil & water conservation at basin scale.

The findings are interesting from the perspective of integrated water management, but also from the perspective of food production and climate change. All over the world investments are currently being made in small scale rainwater harvesting and soil and water conservation structures to help farmers adapt to climate change. This paper suggests that the cumulative effect of numerous small investments in rainwater harvesting could in semi-arid, waters scarce basins reduce the availability of water in downstream irrigation reservoirs. Especially because the downstream regions of most river basins are crucial for food production (Molden et al. 2001), it is important that the potential externalities of upstream investments in rainwater harvesting are adequately addressed.

Despite the fact that adaptability has a forward looking element, adaptation and adaptability share a connotation of reactivity, which is why adaptation is often juxtaposed to mitigation. Adaptation has obtained a dubious reputation in the field of climate change policy, as adaptation could be seen as something that becomes necessary if the desired option of mitigation fails. However, to see adaptation wholly in this negative light would be erroneous because adaptation is a wise element of any policy mix. This is because, when it comes to climate change, we are dealing with a large amount of uncertainty, implying that even if we succeeded in overcoming the many collective action problems and fully agreed on a mitigation agenda and then completely implemented it, we would still be uncertain as to whether or not it was enough and would need adaptation activities, be it globally or locally .

5 Recommendations

In short, we see a number of important contributions of using an ecosystem-based approach to the food security debate.

First, using an ecosystem based approach requires **much more attention** for the (invisible) contributions of the ecosystem to food production, with explicit attention for monetary and non-monetary welfare impacts and potential trade-offs. Despite the economic importance of ecological functions such as pollination and watershed services, these ecosystem services are rarely taken into account in policy making due to their “hidden” nature.

Second, water increasingly has to be **priced** at its real value to economic processes. The worldwide lack of water pricing in especially irrigation schemes is causing ample inefficiencies in economic and environmental terms. This is partly caused by the fact that economist stress the fact that the water demand is inelastic to its current low price levels, yet at higher price levels price increases will lead to major efficiency gains.

Third, using an ecosystem-based approach implies the use of a **multi-level approach**. Ecosystems produce services at different spatial levels, and the governance of food production requires coordination of policies and actions across spatial scales. Ecosystems services do not follow administrative boundaries and therefore require different policy scales to coordinate their interventions.

Fourth, limited knowledge of the production of ecosystem services and the non-linearities associated with ecosystem use require **adaptive governance** of ecosystem services and food security, with flexible institutions that are able to respond to unpredictability and change. In the water sector, this may require investments in bridging capital between the different departments that are and should be involved in integrated water management at basin scale.

Finally, **experiment** with payments for ecosystem services in integrated supply chain management. The clear value combined with the absence of markets and prices for many ecosystem services, provide the necessary conditions for a market for ecosystem services to emerge. Yet, support may be needed to kick-off some of these innovative approaches and interventions.

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